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Exploring the driving forces of energy consumption and environmental pollution in China's cement industry at the provincial level

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ABSTRACT

Identifying strategies for reducing energy consumption and environmental pollution in China's cement industry requires a comprehensive analysis of the sector on various scales, taking into account, in particular, the heterogeneity of abatement options. We develop a spatial and temporal decomposition analysis to quantitatively examine the driving forces of energy consumption and emissions of carbon dioxide (CO_2) and air pollutant emissions in China's cement industry at the national and provincial scales during the period 2005–2012. The results show that, nationally, due to the rapid growth of cement and clinker production, CO_2 emissions experienced a substantial increase. While the emissions of Sulphur dioxide (SO_2), nitrogen oxides (NOx), and particulate matter (PM) were found to initially decrease (due to stringent air pollution standards), and then rose due to the increase of cement production. At the provincial level, we also observe that the developing regions (e.g. Anhui, Jiangsu, Shandong, and Sichuan) have a large share of total emissions of CO_2 and air pollutants, while the megacities of Beijing, Shanghai, and Tianjin contributed less to the total emissions. From spatial decomposition perspective, the energy intensity and emission factor affect CO_2 emissions largely but have a negative linear relationship primarily in developing regions. The findings in this study can directly be used to narrow down the projection of GHG mitigation and air pollution abatement on economic and technical perspectives and help policy maker to identify priority options for tackling the issues of global climate change and improve regional air quality.

1. Introduction

Cement is the major component of concrete; its production currently requires massive amounts of raw materials such as limestone and non-renewable energy, and has harmful impacts on air quality and climate change. The global cement industry contributes 5–7% of global anthropogenic CO_2 emissions (Wang et al., 2013). China has been the largest cement producer in the world since 1985, accounting for ~60% of the global cement production in 2012 (Chen et al., 2015; Edwards, 2013; Zhang et al., 2015a). In recent years, China's cement industry has skyrocketed owing to accelerating economic development and urbanization. The Chinese cement production increased from 210 million ton (Mt) in 1990 to 2210 Mt in 2012, expanding 10 times. Over the same period the associated energy consumption increased by a factor of seven approximately, as result of implementing various energy efficiency measures (National Development and Reform Commission of China (2013)). Several approaches have been used to estimate the factors/trajectories of energy consumption and emissions of CO_2 and air pollutants, as well as forecast the future potentials of energy efficiency improvement, CO_2 mitigation, and air pollution abatement in the Chinese Cement industry (Cai et al., 2016; Chen et al., 2015; Feiz et al., 2015; Ke et al., 2013; Lei et al., 2011a,b; Liu et al., 2015; Wen et al., 2015). For example, Li et al. (2014) used life cycle inventory (LCI) to analyze the air and water pollution of China's cement industry and quantify the potential improvement of avoided environmental damages by using energy efficiency technologies and air pollution control options. It is observed that NOx emissions will decline drastically by

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implementing NOx control technologies, such as SNCR whereas SO_2 emissions will increase in the future primarily due to 1) decrease in the cement and clinker products that will lead to less SO_2 capture during the clinker production, 2) use of excess coal in energy efficient technologies (e.g. AQC (Air Quenching Cooler) and SP (Suspension Preheater) furnaces of waste heat recovery generation technology), which will lead to higher SO_2 emissions (Li et al., 2014). Similarly, Chen et al. (2015) employed a hybrid lifecycle assessment method to estimate the air pollutant emissions and related environmental impacts of China's cement industry, and found that 12% of NOx and 26% of particulate matter (PM) of China's national total are emitted from the cement industry. Improving energy efficiency plays a key role in reducing the environmental burden in China's cement industry (Chen et al., 2015). Past empirical studies indicated that, while improved energy intensity, process efficiency (e.g. improvements of grinding process), structural shift, alternative fuel use, and a lower clinker share helped to reduce energy consumption and CO_2 emissions, an increase in production of cement and clinker had the opposite effect (Wang et al., 2013; Xu et al., 2012). Several energy models found that the cement industry of China still has a large potentials in improving energy efficiency and reducing the emissions of CO_2 and air pollutants through implementing best energy efficiency measures, replacing fossil fuels with alternative fuels, and reducing clinker shares (Hasanbeigi et al., 2013b; Ke et al., 2012; Wang et al., 2014a,b).

Furthermore, mounting studies have shown that co-benefits of energy efficiency in developing countries are higher than in developed countries, and these co-benefits can help to overcome a variety of barriers by adopting best available technologies and to increase energy saving potentials that can be achieved cost-effectively (Hasanbeigi et al., 2013a; Tan et al., 2016; Yang et al., 2013; Zhang et al., 2015a). However, issues on regional heterogeneity (e.g., the difference in urbanization, plant scale, kiln types, fuel mix, energy intensity for clinker/cement, and application rates of air pollution control technologies) in China's cement industry and how these factors affect energy consumption and emissions have not been studied so far. Moreover, air quality benefits from energy efficiency and implementation of air pollution control technologies in Chinese cement industry are usually neglected. Therefore, the purpose of this paper is to fill this gap by quantifying the driving forces of energy consumption and emissions of CO_2 and air pollutants at the provincial level in China's cement industry during the years 2005–2012. To meet this objective, a spatial-temporal index decomposition analysis (ST-IDA) model is developed and used to analyze the trajectories and features for energy intensity, greenhouse gas (CO_2), and air pollutants (SO_2, NOx and PM) emission factor in 30 regions of China. We then quantify the contributions of key driving factors in the evolution of CO_2 and air pollutant emissions at the provincial level using the logarithmic mean divisia index (LMDI) decomposition approach. This paper fills the gap on quantifying key driving forces of energy, and the findings can provide useful insights for integrated assessment models to estimate the cost-effective potentials of energy and emission savings on a regional scale.

The structure of this paper is arranged as follows: Section 2 briefly overviews the cement manufacturing process and energy and emission trajectories in China's cement industry. Section 3 describes the spatial-temporal decomposition analysis and data source in details. Section 4 discusses the spatial and temporal decomposition results for key driving forces of energy consumption and emissions of CO_2 and air pollutants. Finally, we draw conclusions in Section 5.

2. Overview of the cement industry in China

2.1. Cement manufacturing process and its energy use in China

Cement production is a set of energy-intensive processes including raw materials preparation, fuel preparation, and finish grinding (Benhelal et al., 2013). Raw materials (e.g. limestone, chalk and clay) are selected, crushed, ground, and proportioned to the pyro-processing systems' requirements. Here, a jaw/gyratory crusher and a roller/hammer mill are used to crush the limestone (Martin et al., 1999). Around 24% of total process electricity requirements (24–27 kW h/t cement) is consumed during this step (Dai and Xiong, 2013). Next, large kilns are used to produce cement clinker. In this process, raw materials are heated up to 1400–1500 °C to form clinker that is the most energy-intensive process (Ruth et al., 2015; Wang et al., 2013). More specifically, calcium carbonate in limestone dissociates into carbon dioxide and calcium oxide. Currently, the new suspension preheater with a precalciner (NSP) kiln is the most efficient kiln type. Grate or suspension preheaters are used to capture the kiln exhaust gases and heat the raw material before entering the kiln (Murray, 2008). In China, the NSP kiln accounts for 92% of total clinker production in 2012 (China Cement Association, 2010). The specific energy consumption of NSP kiln in China is 3.37 GJ/t clinker, 16–20% lower than that of the shaft kiln (Wen et al., 2015; Zhang et al., 2015a). Once the clinker is formed it is cooled rapidly by the grate cooler or the tube or planetary cooler. After cooling, the clinker is ground together with additives to produce cement (Worrell and Christina, 2004; Habert et al., 2010; Hendriks et al., 2003; Worrell and Christina, 2008; Worrell et al., 2013). The electricity consumption for the raw meal and finish grinding depends on the hardness of the raw materials, the amount of additives, and the desired fineness of the cement. This process consumes the highest amount of electricity in cement production, accounting for 38% of total electricity consumption in China's cement industry (Dai and Xiong, 2013).

2.2. Emissions of CO_2 and air pollutants in China's cement industry

Cement manufacturing is a key CO_2 emitting industry, contributing to 7% of the global anthropogenic CO_2 emissions. The CO_2 emissions in China's cement industry more than doubled from 591 Mt in 2000 to 1380 Mt in 2010, while the cement production tripled during the same period. Generally, emissions of CO_2 in the cement industry stem directly from the combustion of fossil fuels, indirectly from electricity consumption but also from the calcination of the limestone into calcium oxide. The latter is known as process emissions. In China, approximately half of total on-site generated emissions are from limestone calcination and 48% are from fuel combustion. The indirect CO_2 emissions are from consumption of electricity, which is primarily generated in coal-fired power plants, accounting for 9% of the total CO_2 emissions (Ali et al., 2011). The total amount of CO_2 emissions depends on the demand of cement, the types of kilns used, the fuel mix used for clinker burning and power generation, the efficiency of energy utilization, and the clinker/cement ratio. Between 2000 and 2010, the CO_2 intensity of cement dropped by 13% per year on average, due to the declining share of clinker to cement from 0.77 to 0.58 and the decreasing energy intensity of cement production by 3.4% per year during the same period (China Cement Association, 2016; Zhang et al., 2015a).

The emissions of air pollutants in cement industry are not only proportional to the activity level of cement and clinker, but also depend on the kiln type, application of air pollution control technologies, energy consumption and the associated fuel mix. Note that the SO_2 process emissions are negative, mainly because 70% of

SO_2 is absorbed in the process of pyro-processing (Lei et al., 2011a,b); cement products have higher sulfur components (80%) than the raw materials (Cofala and Syri, 1998). China's cement industry is facing challenges in emitting a large amount of various air pollutants, accounting for 4% of SO_2, 10% of NOx, and 15–30% of Particulate Matter (PM), of the total emissions in China (Zhang et al., 2015b). Furthermore, fine particulate ($PM_{2.5}$) emissions from cement kilns varied largely among different provinces. For example, the cement industry in Hunan, Guangxi, Fujian, Qinghai, Chongqing and Jiangxi provinces contributes to more than one-third of the provincial total PM emissions (Trucost and ICCS, 2015; Zhang et al., 2016a,b).

3. Methodology

3.1. Spatial-temporal decomposition analysis

Decomposition analysis is a widely used methodology to explore and quantify the contributions of individual factors in aggregate outcomes, such as energy consumption, emissions and intensity, as well as other environmental indicators (Ang et al., 2003; Cansino et al., 2015; Jeong and Kim, 2013; Liu et al., 2007; Wang et al., 2014a; Wang et al., 2011; Zhao and Chen, 2014). Decomposition analysis can be classified by a variety of different features, such as the study scope, objective, and methodological approach. Xu and Ang (2014) for example, grouped the decomposition analysis into single-level decomposition model and multi-level decomposition model (including multilevel-parallel (M-P) model and multilevel-hierarchical (M-H) model), and they concluded that the multilevel decomposition model could overcome the shortcomings of single-level decomposition analysis, since the former could provide further decomposition to analyze the sub-effect at a finer level (Xu and Ang, 2014). Xu et al. (2012) observed that the growth of cement production in China plays a key role in increasing energy consumption of China's cement industry. While clinker share decline and structural shifts had negative effects on increasing energy consumption in the Chinese cement industry (Xu et al., 2012). Previous decomposition analyses mainly focused on the analysis of the historical trend of one region's performance over time (Ren et al., 2014; Shao et al., 2016; Wang et al., 2014a,b). More recently, however, decomposition analyses have started to include spatial differences between regions and temporal developments in an individual region (Ang et al., 2015; Su and Ang, 2016). Representative example of the extended LMDI analysis is the study by Zhang et al. (2016a,b), which analyzed the driving factors of carbon emission factor in Chinese provinces and found that energy intensity played a vital role in carbon emission factor, followed by carbon emission factor and the share of secondary and tertiary industries (Zhang et al., 2016a,b).

Ding and Li (2017) further employed the extended LMDI to examine the link between energy-related CO_2 emissions and socio-economic driving factors (e.g. urbanization, economic development, and social transition) and the changes in provincial CO_2 emissions, and found that economic development is the largest driver of provincial CO_2 emissions, followed by structural change, energy intensity and social transition. They also found that the contribution of urbanization to CO_2 emissions has large variations among provinces (Ding and Li, 2017). Branger and Quirion (2015) found that CO_2 emission reductions are partly attributed to the technological improvements (i.e. to decrease the clinker share and increase in alternative fuels) and clinker trade, while the activity effect significantly contributes to the emission increase (Branger and Quirion, 2015). With growing interest in spatial and temporal analysis and considering the difference of performance assessment, Ang et al. (2016) proposed three broad categories of decomposition analysis, namely single-country temporal analysis, multi-country temporal analysis, and cross-country analysis, while the spatial index decomposition analysis models are further divided into bilateral-region, radial-region, and multi-region models (Ang et al., 2016). The aim of spatial and temporal analysis is to better understand regional heterogeneity (e.g., the differences in population, economic development level, urbanization process, and energy efficiency) and for regional projections into the future. However, none of these studies explore drivers of energy consumption and emissions in China's cement industry on a provincial scale. Zhang et al., suggested that regional heterogeneity in China's cement industry plays an important role for taking future actions (Zhang et al., 2015b, 2016a,b). The spatial and temporal analysis in this study can be used to quantify the driving forces of energy consumption, CO_2 and air pollutant emissions in China's cement industry at the provincial level. We first analyze the variations and changes of different regions' performance in energy intensity, CO_2 and air pollutant (including SO_2, NO_x and PM) emission factor in comparison with the benchmark region through the spatial-temporal index decomposition analysis (ST-IDA) approach, to identify the major contributing factors and reveal the scope for further improvement. Next, we rank the energy consumption and emission performance among 30 regions in China, and quantified the key driving forces of CO_2 and air pollutant emissions from 2006 to 2012 on the provincial level using LMDI decomposition approach, to illustrate the energy saving and emission mitigation potentials in the future.

The energy consumption (E_n) in cement production is the sum of direct fuel (primarily coal) combustion (E_f), mainly in clinker producing process, and electricity consumption (E_{el}) in grinding process.

$$E_n = E_f + E_{el} \tag{1}$$

The total energy consumption in the China's cement industry (E_n) can be decomposed as:

$$\begin{aligned} E_n &= \sum_{Ri}\left(E_{Ri,f} + E_{Ri,el}\right) = \sum\left(P_{Ri}^{cem}\frac{P_{Ri,f}^{cl}}{P_{Ri}^{cem}}\frac{E_{Ri,}}{P_{Ri}^{cl}} + P_{Ri}^{cem}\frac{E_{Ri,el}}{P_{Ri}^{cem}}\right) \\ &= \sum P_{Ri}^{cem}\left(\beta_{Ri}EI_{Ri,f} + EI_{Ri,el}\right) \end{aligned} \tag{2}$$

Where E_{Ri} represents the energy consumption in Ri^{th} province, P_{Ri}^{cem} the cement production in Ri^{th} province, β_{Ri} the ratio of clinker to cement in Ri^{th} province and $EI_{Ri,f}$ the energy intensity in Ri^{th} province.

The energy intensity (EI_i) in cement production in Ri^{th} province can be calculated as:

$$EI_i = \frac{E_i}{P_{Ri}^{cem}} = \left(\beta_{Ri}EI_{Ri,f,clinker} + EI_{Ri,el}\right) \tag{3}$$

Generally, the emissions (Em) of CO_2, SO_2, NOx, and PM can be split into three categories: emissions from fuel combustion (f), process emissions (p), and indirect emissions from electricity consumption (el). For CO_2 emission, we split the total emissions into three parts:

$$Em = Em_f + Em_p + Em_{el} \tag{4}$$

The total emissions for fuel combustion in Ri^{th} province are decomposed as follow:

$$Em_{Ri,f} = P_{Ri}^{cem}\frac{P_{Ri}^{cl}}{P_{Ri}^{cem}}\frac{E_{Ri,f}}{P_{Ri}^{cl}}\frac{Em_{f,Ri}}{E_{Ri}} = P_{Ri}\beta_{Ri}EI_{Ri,f}EF_{Ri,f} \quad (5)$$

Where E_{Ri} is the fuel combustion in Rith province, $E_{Ri,f}$ is the energy consumption by fuel types in Rith province, β_{Ri} is the rate of clinker to cement in Rith province, and $EF_{Ri,f}$ is the emission factor of the fuel mix in Rith province.

The process emissions Em_p are given as:

$$Em_{p,Ri} = P_{Ri}^{cem}\frac{P_{Ri}^{cl}}{P_{Ri}^{cem}}\frac{Em_{p,Ri}}{P_{Ri}^{cl}} = P_{Ri}\beta_{Ri}EF_{Ri,p} \quad (6)$$

Where $EF_{Ri,p}$ is the process emission factor in Rith province.

The indirect emission from electricity consumption can be estimated as:

$$Em_{el} = \sum Em_{Ri,el} = P_{Ri}^{cem}\frac{E_{Ri,el}}{P_{Ri}^{cem}}\frac{Em_{Ri,el}}{E_{Ri,el}} = P_{Ri}EI_{Ri,f}EF_{Ri,el} \quad (7)$$

Where $EF_{el,Ri}$ is the electricity consumption in Rith province, $EI_{Ri,f}$ is the electricity intensity for cement production and $EF_{Ri,el}$ is the indirect emission factor from electricity use in Rith province.

Therefore the total CO_2 emissions of cement industry in China can be given as:

$$E_m = \sum E_{m,\ Ri} = \sum P_{Ri}*\left(\left(EI_{Ri,f}\beta_{Ri}EF_{Ri,f} + EF_{Ri,p}\right) + EI_{Ri,f}EF_{Ri,el}\right) \quad (8)$$

As mentioned in Section 2.2, it is not possible to estimate the amount of air pollutant emissions captured in the process and the amount of emissions directly linked to energy use, because some pollutants (e.g. sulfur) are often absorbed during cement producing process (Cofala and Syri, 1998). Detailed information is not available to distinguish air pollutant (SO_2, NOx and PM) emissions from fuel combustion, process emission and electricity consumption. Therefore, in this study emission factors of air pollutants represent final emissions from the whole cement production process. We simply split the total emission as follows:

$$Em_{pol,Ri} = P_{Ri}^{cem}\frac{E_{Ri}}{P_{Ri}^{cem}}\frac{Em_{pol,Ri}}{E_{Ri}} = P_{Ri}EI_{Ri}EF_{Ri,p} \quad (9)$$

Where $Em_{pol,Ri}$ is the total emission of pollutants (SO_2, NOx or PM) in Rith province, and E_{Ri} is the total energy consumption of cement production in Rith province. Based on Equation (9), the emission factor of one pollutant is expressed as:

$$EmI_{j,Ri} = \frac{Em_{pol,Ri}}{P_{Ri}^{cem}} = \frac{E_{Ri}}{P_{Ri}^{cem}}\frac{Em_{pol,Ri}}{E_{Ri}} = EI_{Ri}EF_{Ri,p} \quad (10)$$

Where $EmI_{j,Ri}$ is the emission factor of jth pollutant in Rith province.

3.1.1. Multi-regional spatial-temporal decomposition analysis

The multi-regional spatial-temporal decomposition analysis can provide a comparison/benchmarking between each target region in different years and the group average (Ang et al., 2016, 2015; Su and Ang, 2016). As shown is Fig. 1, the solid lines represent the direct comparisons between each of the Rith province and the national average, while the dashed lines represent the indirect decomposition results. The difference between two target provinces (indirect calculations) can be calculated from the direct decomposition results (e.g. energy consumption) for the two provinces and the national average as follows:

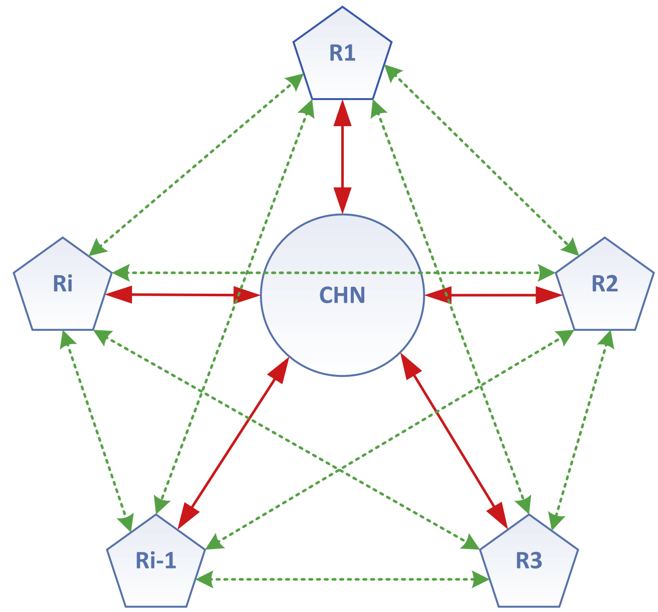

Fig. 1. The multi-regional spatial decomposition analysis.

$$\begin{aligned}\Delta E_{(R1-R2)} &= \Delta E_{(R1-CHN)} - \Delta E_{(R2-CHN)} \\ &= (E_{R1} - E_{CHN}) - (E_{R2} - E_{CHN})\end{aligned}$$

where $\Delta E_{(R1-R2)}$ is the indirect result for the difference between any two provinces, $\Delta E_{(R1-CHN)}$ and $\Delta E_{(R2-CHN)}$ are the direct results for the difference between province R_1 and the national average and between province R_2 and the national average, respectively. E_{R1} represents the energy intensity in province R_1 and E_{CHN} the national average energy intensity.

In this study, the energy intensity, and the CO_2, SO_2, NOx and PM emission intensities are decomposed with the spatial-temporal index decomposition analysis (ST-IDA), based on Equations (3)–(10). We select 2005, 2009 and 2012 as the study periods, and the benchmark region is constructed from the average of all provinces in these years to present a fair picture of energy intensity and emission factor from 2005 to 2012.

Compared with the benchmark region (CHN), the energy intensity difference in one region (e.g., R1) is decomposed into three parts, i.e. the clinker energy intensity effect, clinker share effect, and electricity intensity effect. According to the guidelines of LMDI approach, the additive LMDI method is best (Ang, 2016, 2015; Ang et al., 2003), in the study, and the corresponding effects can be calculated as:

$$\begin{aligned}\Delta\widetilde{EI}_{clinker,\ (Ri-CHN)} &= \left(\widetilde{EI}_{clinker,\ Ri} - \widetilde{EI}_{clinker,\ CHN}\right) \\ &= \frac{(EI_{Ri} - EI_{CHN})}{\ln EI_{Ri} - \ln EI_{CHN}} \ln\frac{\widetilde{EI}_{clinker,\ R1}}{\widetilde{EI}_{clinker,CHN}}\end{aligned} \quad (12)$$

$$\Delta\tilde{\beta}_{(Ri-CHN)} = \left(\tilde{\beta}_{Ri} - \tilde{\beta}_{CHN}\right) = \frac{(EI_{Ri} - EI_{CHN})}{\ln E_{Ri} - \ln E_{CHN}} \ln\frac{\tilde{\beta}_{R1}}{\tilde{\beta}_{CHN}} \quad (13)$$

$$\Delta EI_{el,(Ri-CHN)} = \left(EI_{el,\ Ri} - EI_{el,\ CHN}\right) \quad (14)$$

The production structure and energy intensity effects between two provinces can be calculated as:

$$\Delta\widetilde{EI}_{(R1-R2)} = \widetilde{EI}_{R1} - \widetilde{EI}_{R2} = \left(\widetilde{EI}_{R1} - \widetilde{EI}_{CHN}\right) - \left(\widetilde{EI}_{R2} - \widetilde{EI}_{CHN}\right) \tag{15}$$

$$\Delta\tilde{\beta}_{(R1-R2)} = \tilde{\beta}_{R1} - \tilde{\beta}_{R2} = \left(\tilde{\beta}_{R1} - \tilde{\beta}_{CHN}\right) - \left(\tilde{\beta}_{R1} - \tilde{\beta}_{CHN}\right) \tag{16}$$

Compared with the benchmark region, the emission intensities of CO_2, SO_2, NOx and PM are decomposed with ST-IDA model into two factors, energy intensity effect, and emission factor effects. The CO_2 emission factor is an aggregated emission factor including emissions from coal burning, process emissions due to the chemical reaction from limestone calcination, and indirect emissions from electricity consumption:

$$\Delta\widetilde{EI}_{(Ri-CHN)} = \left(\widetilde{EI}_{Ri} - \widetilde{EI}_{CHN}\right) = \frac{(EmI_{Ri} - EmI_{CHN})}{\ln EmI_{Ri} - \ln EmI_{CHN}} \ln \frac{\widetilde{EI}_{Ri}}{\widetilde{EI}_{CHN}} \tag{17}$$

$$=$$

$$\Delta\widetilde{EF}_{(Ri-CHN)} = \left(\widetilde{EF}_{Ri} - \widetilde{EF}_{CHN}\right) = \frac{(EmI_{Ri} - EmI_{CHN})}{\ln EmI_{Ri} - \ln EmI_{CHN}} \ln \frac{\widetilde{EF}_{Ri}}{\widetilde{EF}_{CHN}} \tag{18}$$

Furthermore, equations (12)–(19) will be employed to analyze the spatial-temporal decomposition results of energy intensity and emission factors of CO_2, SO_2, NOx, and PM across provinces.

3.1.2. Temporal decomposition analysis

The multi-region decomposition approach provides the spatial decomposition analysis results between one region and a benchmark region, and further between any two regions. In order to quantify and analyze the temporal changes of different driving forces to the energy consumption and emissions on the provincial level, we carried out temporal decomposition analysis for each province and China altogether with the LMDI approach (Ang, 2016, 2015; Ang et al., 2003). The LMDI method has been used to quantitatively examine the impact of different factors on the changes of energy use and emissions, due to its advantages of adaptability, independency, and lack of a residual term (Fujii et al., 2013; Wang et al., 2013). Currently, the LMDI method can be distinguished in two different types. First, used in a period-wise manner it is used to focus on analyzing the factors between the first and the last year, while the time-series analysis compares the key factors year by year. The time-series analysis of LMDI has often been used to figure out the developing trends and the historical impact of different factors during the study period (Wang et al., 2013). For example, Lin and Long (2016) conducted the study on driving factors of CO_2 emission changes in China's chemical industry, based on time series data and provincial panel data, and found that energy intensity reductions and energy structure have positive roles in reducing CO_2 emissions, while output per worker and industrial economic scale could increase emissions. Furthermore, they also concluded that the above driving factors differ widely across provinces, which provided insightful suggestions for regional policymakers in designing energy and carbon policies (Lin and Long, 2016). Furthermore, they also concluded that the above driving factors differ widely across provinces, which provided insightful suggestions for regional policymakers in designing energy and carbon policies (Lin and Long, 2016). As shown in the literature, the LMDI has been used mainly in energy and GHG emission studies. Due to the possibility of using LMDI to analyze the key factors of pollution in single industry, the time-series analysis of LMDI are used to clarify the driving forces of China's cement industry which is one novelty of this paper. We choose the 2005 as the base year (year 0). The effects from factors including cement production (including △coal-P, △pro-P, and △ele-P), clinker share in cement (△coal-β), energy intensity (including △coal-EI, and △ele-P), and emissions factors (including △coal-EF, △pro-EF, and △ele-EF) are quantified by Equations (19)–(27).

$$\Delta coal - P = \Delta E^{P,f}_{CO2,\ Ri,t} = \frac{\left(E^f_{CO2,Ri,t} - E^f_{CO2,Ri,0}\right)}{\ln E^f_{CO2,Ri,t} - \ln E^f_{CO2,Ri,0}} \ln\left(\frac{P^{cem}_{Ri,t}}{P^{cem}_{Ri,0}}\right) \tag{19}$$

$$\Delta coal - \beta = \Delta E^{\beta,f}_{CO2,\ Ri,t} = \frac{\left(E^f_{CO2,Ri,t} - E^f_{CO2,Ri,0}\right)}{\ln E^f_{CO2,Ri,t} - \ln E^f_{CO2,Ri,0}} \ln\left(\frac{\beta_{Ri,t}}{\beta_{Ri,0}}\right) \tag{20}$$

$$\Delta coal - EI = \Delta E^{EI,f}_{CO2,\ Ri,t} = \frac{\left(E^f_{CO2,Ri,t} - E^f_{CO2,Ri,0}\right)}{\ln E^f_{CO2,Ri,t} - \ln E^f_{CO2,Ri,0}} \ln\left(\frac{EI^f_{Ri,t}}{EI^f_{Ri,0}}\right) \tag{21}$$

$$\Delta coal - EF = \Delta E^{EF,f}_{CO2,\ Ri,t} = \frac{\left(E^f_{CO2,Ri,t} - E^f_{CO2,Ri,0}\right)}{\ln E^f_{CO2,Ri,t} - \ln E^f_{CO2,Ri,0}} \ln\left(\frac{EF^f_{Ri,t}}{EF^f_{Ri,0}}\right) \tag{22}$$

$$\Delta pro - P = \Delta E^{P,pro}_{CO2,\ Ri,t} = \frac{\left(E^{pro}_{CO2,Ri,t} - E^{pro}_{CO2,Ri,0}\right)}{\ln E^{pro}_{CO2,Ri,t} - \ln E^{pro}_{CO2,Ri,0}} \ln\left(\frac{P^{cem}_{Ri,t}}{P^{cem}_{Ri,0}}\right) \tag{23}$$

$$\Delta pro - EF = \Delta E^{EF,pro}_{CO2,\ Ri,t} = \frac{\left(E^{pro}_{CO2,Ri,t} - E^{pro}_{CO2,Ri,0}\right)}{\ln E^{pro}_{CO2,Ri,t} - \ln E^{pro}_{CO2,Ri,0}} \ln\left(\frac{EF^{pro}_{Ri,t}}{EF^{pro}_{Ri,0}}\right) \tag{24}$$

$$\Delta ele - P = \Delta E^{P,el}_{CO2,\ Ri,t} = \frac{\left(E^{el}_{CO2,Ri,t} - E^{el}_{CO2,Ri,0}\right)}{\ln E^{el}_{CO2,Ri,t} - \ln E^{el}_{CO2,Ri,0}} \ln\left(\frac{P^{cem}_{Ri,t}}{P^{cem}_{Ri,0}}\right) \tag{25}$$

$$\Delta ele - EI = \Delta E^{EI,el}_{CO2,\ Ri,t} = \frac{\left(E^{el}_{CO2,Ri,t} - E^{el}_{CO2,Ri,0}\right)}{\ln E^{el}_{CO2,Ri,t} - \ln E^{el}_{CO2,Ri,0}} \ln\left(\frac{EI^{el}_{Ri,t}}{EI^{el}_{Ri,0}}\right) \tag{26}$$

$$\Delta ele - EF = \Delta E^{EF,el}_{CO2,\ Ri,t} = \frac{\left(E^{el}_{CO2,Ri,t} - E^{el}_{CO2,Ri,0}\right)}{\ln E^{el}_{CO2,Ri,t} - \ln E^{el}_{CO2,Ri,0}} \ln\left(\frac{EF^{el}_{Ri,t}}{EF^{el}_{Ri,0}}\right) \tag{27}$$

The effects from cement production (△P), energy intensity (△EI), and emission factors (△EF) on the changes of PM, SO_2 and NOx emissions are quantified by Equations (28)–(30):

$$\Delta P = \Delta E^{P}_{pol,\ Ri,t} = \frac{\left(Em_{pol,Ri,t} - Em_{pol,Ri,0}\right)}{\ln Em_{pol,Ri,t} - \ln Em_{pol,Ri,0}} \ln\left(\frac{P^{cem}_{Ri,t}}{P^{cem}_{Ri,0}}\right) \tag{28}$$

$$\Delta EI = \Delta E^{EI}_{pol,\ Ri,t} = \frac{\left(Em_{pol,Ri,t} - Em_{pol,Ri,0}\right)}{\ln Em_{pol,Ri,t} - \ln Em_{pol,Ri,0}} \ln\left(\frac{EI_{Ri,t}}{EI_{Ri,0}}\right) \quad (29)$$

$$\Delta EF = \Delta E^{EF}_{pol,\ Ri,t} = \frac{\left(Em_{pol,Ri,t} - Em_{pol,Ri,0}\right)}{\ln Em_{pol,Ri,t} - \ln Em_{pol,Ri,0}} \ln\left(\frac{EF_{pol,Ri,t}}{EF_{pol,Ri,0}}\right) \quad (30)$$

3.2. Data sources

The historical cement and clinker output of each province are obtained from various issues of the China statistical year book (National Bureau of Statistics of China, 2013, 2011), China cement almanac (China Cement Association, 2010), and China Cement Association (China Cement Association, 2012). The historical coal combustion and electricity consumption data in cement industry of each province are from China energy statistical yearbook (National Bureau of Statistics of China (2013), 2011), China cement almanac (China Cement Association, 2010), and existing studies (China Building Materials Academy and Energy saving center of Sichuan Province (2010); Hasanbeigi et al., 2013a, 2010; Zhang et al., 2015b).

The provincial CO_2 emission data in cement industry are compiled based on the corresponding cement production, coal combustion, electricity consumption, and emission factors. The CO_2 emission factors for coal are taken from Lawrence Berkeley National Laboratory (LBNL) (Hasanbeigi et al., 2013b; Ke et al., 2012). The CO_2 emission factors for electricity by each province are taken from regional grid baseline emission factors of China (National Center for Climate Change Strategy and International Cooperation of China (2010)). The CO_2 emission factors for processes are from 2011 Cement Sustainability Initiative (CSI) on CO_2 and Energy Accounting and Reporting Standard for the Cement Industry (World Business Council for Sustainable Development, 2011).

The provincial emission inventories of air pollutants (including SO_2, NOx and PM) for cement industry are developed based on energy data and related emission coefficients. Due to the data constraints, the emission coefficients of NOx are obtained from IPCC (Intergovernmental Panel on Climate Change, 2014). The emission coefficients of PM are from the World Business Council for Sustainable Development (WBSCD) (World Business Council for Sustainable Development, 2013), the emission coefficients of SO_2 are taken from Ministry of Environmental Protection of China (Ministry of Environmental Protection of China (2013)), and relevant literature surveys (Lei et al., 2011a,b; Zhang et al., 2015b).

Note that several different data sources are not always completely consistent due to differences in the system boundaries and definitions. For example, some studies did not take into account the indirect carbon emissions from electricity (Ke et al., 2013). Hence, outlier values were identified and revised by considering the local realities and in some cases by communication with experts. Moreover, we assume that the emission coefficients remain the same over time.

4. Results and discussion

In this section, we first shape the trajectories and features for energy intensity, greenhouse gas (CO_2), and air pollutant (SO_2, NOx and PM) emission factor among 30 regions in China with the spatial-temporal index decomposition analysis (ST-IDA) model. We then quantify contributions of key driving factors in the evolution of CO_2 and air pollutant emissions at the provincial level using the LMDI decomposition approach, and discuss their energy saving and emission mitigation potentials in the cement industry in the future.

4.1. Spatial differences in energy intensity by province in China

In this study, differences between the provincial energy intensity and the benchmark region are quantified with the ST-IDA model. Energy intensity instead of energy consumption is chosen as the indicator. This is because energy consumption is scale-dependent while energy intensity is not; therefore it represents a more appropriate indicator for comparisons among 30 different provinces of China.

We take the national average of energy intensity as a benchmark as discussed in Section 3.1.1, that is keep constant during the study period (2005–2012). The overall energy intensity in China's cement production was 2.59 GJ/t, while the thermal energy intensity of clinker and the electricity intensity of cement were 3.55 and 0.28 GJ/t, respectively. Variations in energy intensity across the provinces primarily arise from three factors, viz. The clinker energy intensity, the clinker share, and the electricity intensity. Since clinker energy intensity and output structure are the dominant factors in driving the energy intensity variations among different regions, and the contributions from electricity intensity effects are relatively not so important, therefore, at this point we only discuss the interactions between the effects from energy intensity and output structure. Fig. 2 shows the calculated clinker energy intensity (Y-axis) and production structure (representing the clinker share of cement production) effects (X-axis). Each dot denotes one province and different colors denote cement productions levels (Mt) in each year. The dots are numbered by ranking the cement

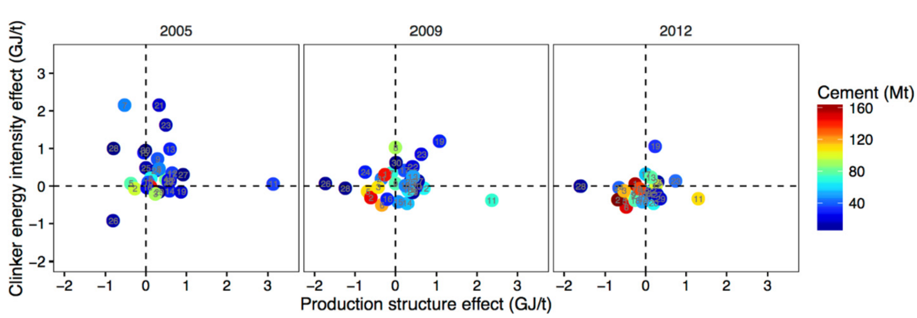

Fig. 2. Decomposition analysis of energy intensity in cement industry for 30 provinces in China. Note: (1) each number denotes the name of each province ordered by the cement production in 2005 from the highest to the lowest: 1-Shandong; 2-Jiangsu; 3-Zhejiang; 4-Guangdong; 5-Hebei; 6-Henan; 7-Hebei; 8-Sichuan; 9-Hunan; 10-Jiangxi; 11-Anhui; 12-Guangxi; 13-Yunnan; 14-Fujian; 15-Liaoning; 16-Shanxi; 17-Chongqing; 18-Shaanxi; 19-Jilin; 20-Guizhou; 21-Inner Mongolia; 22-Gansu; 23-Xinjiang; 24-Heilongjiang; 25-Beijing; 26-Shanghai; 27-Ningxia; 28-Tianjin; 29-Hainan; 30-Qinghai. (2) The color of the scatter dots denotes the cement production of each province in each year.

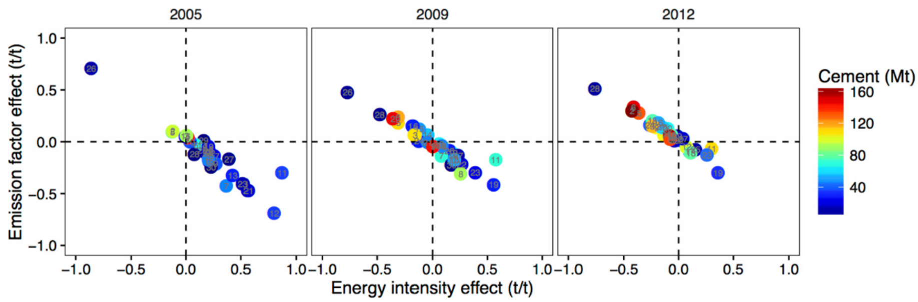

Fig. 3. Decomposition analysis of CO2 emission factor in cement industry for 30 provinces in China using the ST-IDA model. Note: the same with Fig. 1.

production in 2005, from the highest to the lowest. The distribution pattern of the panel plots indicates changes of drivers in 2005, 2005 to 2009, and 2009 to 2012. In Fig. 2, a positive value for the clinker energy intensity effect indicates that one region is less efficient in energy use for cement production than the three-year national average, while a positive value for the production structure effect indicates that one region is more intensive in energy use than the national average.

In Fig. 2, the plotting area is divided into four quadrants by two dashed lines representing the clinker energy intensity effect and production structure effect to be zero respectively, and (0, 0) represents the benchmark region with energy intensity at 2.59 GJ/t. In 2005, 73% of the provinces are in the upper quadrants, showing positive clinker energy intensity effects compared with the benchmark region, and the magnitude varied widely from 0.10 GJ/t in Jiangxi and 2.15 GJ/t in Inner Mongolia, indicating divergent energy intensity for clinker production among regions due to different technologies applied in clinker production. In addition, in 2005, 76% of the provinces are located in the right quadrants, showing positive production structure effects, and magnitude is within 1 GJ/t except for Anhui (3.13 GJ/t), indicating a higher rate of clinker to cement resulted in higher energy intensity.

Between 2005 and 2012, the cement production in China increased from 1067 Mt to 2210 Mt, i.e. with an annual average growth rates of 29%. The distribution of the provinces developed from a divergent pattern to a more centralized pattern, and both drivers moved from the positive effects towards negative effects, which resulted in significant improvements in energy intensity from 3.13 GJ/t in 2005 to 2.66 GJ/t in 2009, and 2.26 GJ/t cement in 2012. In some regions, the production structure is the vital driving factor for the energy intensity change. For example, Anhui that comprises the largest clinker export has a 60% higher energy intensity than the benchmark region. At the same time, the production structure played an important role in the decline of total energy intensity in Shanghai and Tianjin, because these two cities are clinker importing regions. The results indicate that clinker exporting regions have large potentials to improve energy efficiency than clinker importing regions (Zhang et al., 2015b). In 2012, the clinker energy intensity and production structure together contributed to 57–73% of the overall energy intensity improvement through the country. The cement industry has developed from energy and clinker intensive to a more efficient production pattern, due to the fast development of large-scale NSP kilns and phase-out of inefficient plants (smallest shaft kilns and wet-process with vertical shaft kilns).

4.2. Spatial differences in emissions of CO_2 and air pollutants across provinces in China

In China, CO_2 emission factor for cement production has decreased by 13% between 2005 and 2012, from 0.86 t CO_2/t in 2005 to 0.77 t CO_2/t in 2009 and 0.76 t CO_2/t in 2012. The CO_2 factor of each province is compared with the temporal and spatial average of the 30 regions of 0.79 t CO_2/t cement. Fig. 3 shows the estimated energy intensity (X-axis) and CO_2 emission factor effects (Y-axis). A positive value for the energy intensity effect indicates that one province is less efficient in energy use for cement production than the three-year national average, while a positive value for the CO_2 emission factor effect indicates that one region is more intensive in CO_2 emission than the national average. As shown in Fig. 3, the results illustrate a negative linear relationship between the two factors for most of the provinces. This is because technologies used to reduce energy use in the cement industry result in more efficient fuel use and coal combustion, and less incomplete combustion by-products. During the study period, the energy intensity and emission factor effect vary greatly across the provinces but much less in time. The main reason is that the volume of clinker production and clinker export declined over the whole period. Between 2009 and 2012, the emission factor effect changes much less than the year 2005 per province, although for Shanghai and Tianjin it is somewhat higher. This indicates that the energy intensity effect had a negative impact on the emission factor effect.[1]. It should be noted that there is no carbon capture and storage (CCS) technology installed in the Chinese cement industry and that the process CO_2 emission factor remained almost the same in the whole study period (2007–2012). Hence, to keep cement quality constant, improving energy efficiency and increasing the use of low carbon fuels is an important strategy to mitigate CO_2 emissions in the short term, while the CCS would have large contribution to mitigating CO_2 emissions in the long-term period (Wen et al., 2015). Note that application of CCS can increase the energy use, ceteris paribus, by 8–14% (Findlay et al., 2009).

We have used the same method to estimate the differences between provincial and national average SO_2, PM and NOx emission factor, which can be decomposed into two factors, energy intensity effect and emission factor effect. At the national scale, the SO_2 emissions in the cement industry have initially slightly declined from 1047 kt in 2005 to 970 kt in 2009, and then increased to 1461 kt in 2012. The main reason for the initial decrease was that a more stringent standard of air pollutant emissions in the cement industry was released and implemented in 2004, which accelerated the implementation of SO_2 control technologies, such as flue-gas desulfurization (FGD). However, after the global economic crisis in 2008, the government increased investment in infrastructure construction, which resulted in large increases in cement production after 2009, and the corresponding SO_2 emissions have increased since then. At the same time, the SO_2 emission factor

[1] Energy efficiency improvement had a positive effect on reductions of total emission intensity.

significantly decreased from 0.98 t/kt in 2005 to 0.61 t/kt cement in 2012, due to the fast development of NSP kilns and the implementation of the air pollutant emission standard of 2004. Compared to the average SO_2 emission factor at 0.67 t/kt cement for the benchmark region, 67%, 20% and 30% of provinces have higher SO_2 emission factor in 2005, 2009 and 2012 respectively, indicating emission reducing potentials for two thirds of the provinces in 2005 and clean production transitions in the cement industry from 2005 to 2009. In 2005, 87% of the provinces show a positive energy intensity effect for SO_2, while the emission factor effect is positive for 50% of the provinces (reference to Fig 4). There are 12 provinces placed in the upper-right quadrant, showing higher energy intensity than the benchmark region driven by both energy intensity and emission factor effects. These 12 provinces also have large potentials in reducing the SO_2 emission factor, if best energy efficiency measures and desulfurization technologies (e.g. wet flue gases desulfurization and Low Sulphur Coal) are adopted (Wang et al., 2016). The first five provinces with the largest emission factor differ widely compared to the benchmark region and appear all in the upper-right quadrant, i.e. Chongqing (3.5 t/kt), Guangxi (2.9 t/kt), Hunan (1.8 t/kt), Shaanxi (1.11 t/kt) and Sichuan (0.9 t/kt).

The trajectories of PM emissions were found to be similar to those of SO_2 emissions, which first experienced a decrease from 2005 (5716 kt) to 2009 (3656 kt), and then an increase from 2009 to 2012 (5348 kt), due to the more stringent emission standard and subsequent increased investment in infrastructure constructions since 2009. Correspondingly, on national average, the PM emission factor dramatically decreased from 5.3 t/kt in 2005 to 2.2 t/kt in 2009, and then slightly increased to 2.4 t/kt in 2012, indicating the effectiveness of the new emission standard since 2004 and possibly less strict emission controls after the economic crisis after 2008. Overall, the changes in emission intensities and emission factors had similar effects for PM emissions with that for SO_2 emissions. Compared to the spatial and temporal average PM emission factor of 3.0 t/kt in the benchmark region, 70%, 33% and 33% of the provinces have higher PM emission factor in 2005, 2009 and 2012 respectively. Particularly in 2005, 60% of provinces were in the upper-right quadrant, means that both energy intensity and emission factor effect contribute the increase in PM emission factor. The emission factor effect varied significantly among provinces, ranging from −2.1 t/kt in Shanghai to 13.6 t/kt in Hunan, indicating very distinct effectiveness in PM emission controls in different provinces. Due to energy efficiency improvement and more stringent end-of-pipe emission control measures, significant PM emission factor reductions occurred during 2005–2009. Indeed, the same feature was also found in other Chinese industries during the same period (Lyu et al., 2016). As illustrated in Fig. 4, from 2005 to 2009, the scatters changed from dispersed to centralized, indicating a process of higher efficiency and better emission control in cement production. In 2009, 63% of the provinces are in the right quadrants, and 27% of the provinces are in the upper quadrants, showing that energy intensity has large effects on the PM emission factor. From 2009 to 2012, the provinces had a more centralized distribution around (0, 0), with 70% and 63% of the provinces experiencing negative effects from energy intensity and emission factor, respectively. The main reason is that more efficient PM control technologies been implemented homogeneously across the country.

Different from SO_2 and PM, the decomposition analysis of NOx emission factor for 30 provinces in China showed a more divergent distribution during the study period, indicating large variations in energy efficiency and emission control technologies among different provinces. Over the period of the seven years, the NOx emissions continuously increased from 1344 kt in 2005 to 1689 kt in 2009, and further increased to 2780 kt in 2012, while at the same time, the NOx emission factor did not change much, from 1.26 t/kt in 2005 to 1.03 t/kt in 2009 and 1.26 t/kt in 2012. The main reason is that before 2010 China did not have a mandatory control for NOx emissions. Even though a 10% emission reduction target was set during the 12th Five-Year Plan (FYP), the use of NOx control

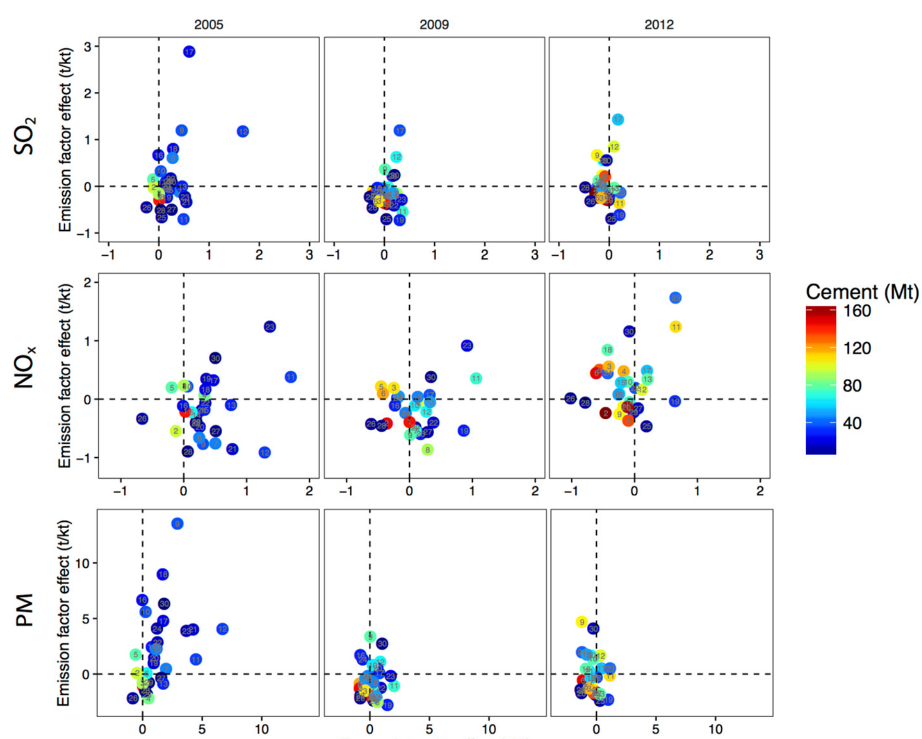

Fig. 4. Decomposition analysis of SO_2, NOx and $PM_{2.5}$ emission factor in cement industry for 30 provinces in China using the ST-IDA model. Note: the same with Fig. 1.

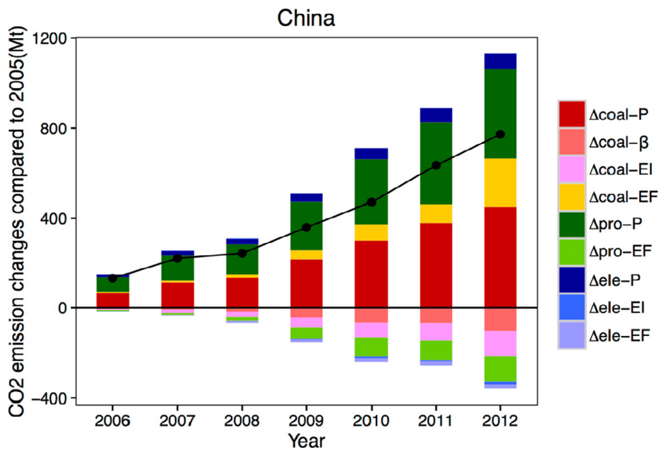

Fig. 5. LMDI decomposition analysis of CO_2 emissions from China's cement industry compared to 2005.

technologies was not enforced before 2012. Compared to the average NOx emission factor of 1.17 t/kt in the benchmark region, 50%, 30% and 47% of provinces had higher emission intensities in 2005, 2009 and 2012 respectively. From the period 2005–2009, 63% of the provinces were in the right quadrants showing that energy intensity effect was an important contributor to the increase in NOx emission factor. For the same years, 67% of provinces were in the lower quadrants, indicating that the emission factor effect plays a dominant role in reducing the NOx emission. In contrast, in 2012, 70% of the provinces were in the left quadrants showing that energy intensity can significantly reduce the amount of the increased NOx emission. At the same time, 60% of provinces were in the upper quadrants, means that emission factor effect significantly contributed towards the increase of NOx emission.

4.3. Changes in CO_2 emissions in the cement industry over time

The trajectories of CO_2 emissions in China's cement industry during the period 2005–2012 are presented in Fig. 5. The CO_2 emissions from coal consumption are decomposed to the effects of cement production (△coal-P), clinker share to cement (△coal-β), energy intensity (△coal-EI) and emission factor (△coal-EF). The CO_2 emissions from the process are decomposed to the effects from emission factors of thermal decomposition of calcium carbonate (△pro-EF) and clinker production (△pro-P). The indirect CO_2 emissions from electricity consumption are decomposed to the effects from cement production (△ele-P), energy intensity (△ele-EI) and emission factor (△ele-EF). The incremental CO_2 emissions increased from 132 Mt in 2006 to 772 Mt in 2012, compared to 2005's level (see the black line of Fig. 5). In general, the activity effect of cement production (△pro-P) and activity effect of coal consumption (△coal-P) are the top two largest positive contributors to the increment of CO_2 emissions, followed by the activity effect of electricity consumption (△ele-P) and emission factor effect of coal combustion (△coal-EF). Compared to 2005, the CO_2 emission from activity effect of coal consumption increased by 65 Mt in 2006, and 450 Mt in 2012 of the cement total, and the contribution from activity effect of cement production to the total CO_2 emission also increased rapidly, which increased by 64 Mt in 2006, and 400 Mt in 2012, respectively. Between 2005 and 2012, the factors of energy intensity for coal combustion (△coal-EI), process emission factors (△pro-EF), and energy intensity for electricity consumption (△ele-EI), by contrast, have negative contributions to the changes of CO_2 emissions. In details, compared to 2005, the energy intensity for coal combustion (△coal-EI) effect reduced CO_2 emissions by 4 Mt in 2006, and 102 Mt in 2012, due to the application of energy efficiency measures. The process emission factors (△pro-EF) reduced CO_2 emissions by 5 Mt in 2006, and 113 Mt in 2012, as a result of shifting process from wet to dry.

Fig. 6 shows the provincial LMDI decomposition analysis of CO_2 emissions from cement industry in 2012 compared to 2005. Generally, the CO_2 emissions across provinces are strongly affected by the scale and nature of cement plants, particularly driven by activity effects of coal consumption (△coal-P) and activity effects of process calcination (△pro-P). The process emissions in Anhui have the largest contribution to CO_2 total emissions in 2012 compared to 2005. The main reason is that Anhui is a large clinker producer. Compared to 2005, the reduction in energy intensity

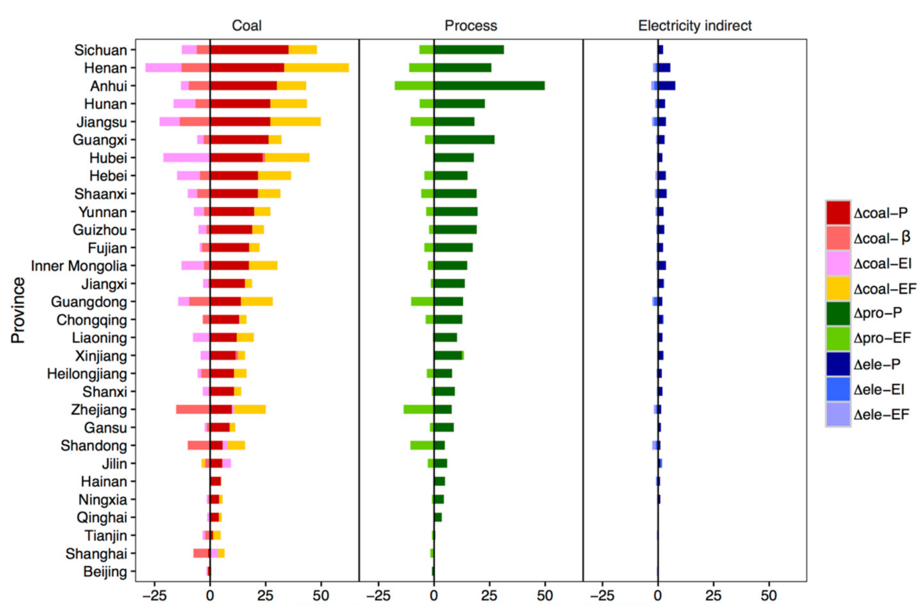

Fig. 6. LMDI decomposition analysis of CO_2 emissions from cement industry in 2012 in 30 provinces compared to 2005. Note: the provinces are ordered by the cement production increment from 2005 to 2012, from the largest increment in Sichuan and the lowest increment in Beijing.

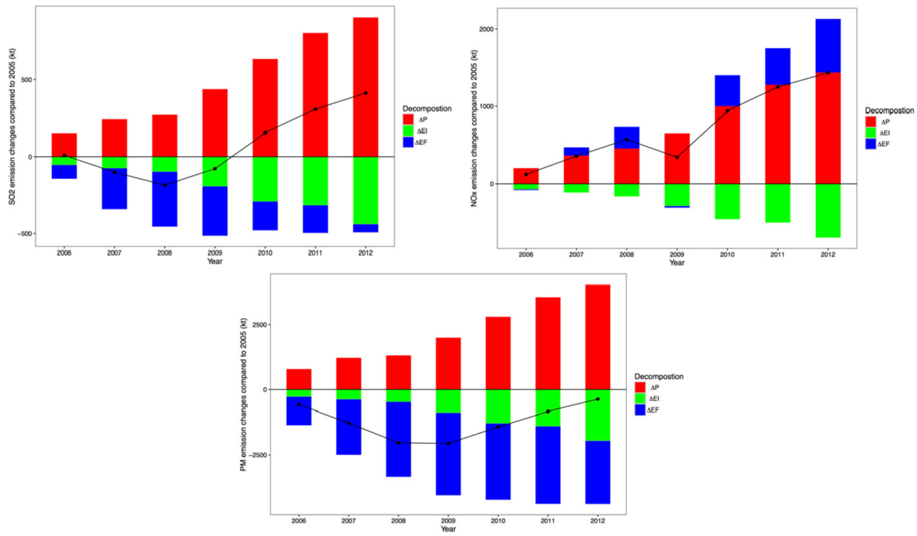

Fig. 7. LMDI decomposition analysis of SO_2, PM and NO_x emissions from China's cement industry compared to 2005.

(including coal combustion (△coal-EI) and electricity consumption (△ele-EI)) in 2012 has significantly reduced CO_2 emissions. The energy intensity effect in Hubei, for example, contributes 51% to the CO_2 emission, followed by Inner Mongolia (42%), Liaoning (34%), and Henan (28%). In developing regions (e.g. Hainan, Ningxia, and Qinghai), coal combustion (△coal-P) and clinker production (△pro-P) do not have strong effects on the CO_2 emission changes. The main reason is that energy efficiency has not been significantly improved in these regions, primarily due to financial limitations.

4.4. Changes in air pollutant emissions in the cement industry

The trajectories of air pollutant emission changes in China's cement industry are decomposed into three factors: activity effect, energy intensity effect, and emission factor effect. Fig. 7 shows that on national level, the emissions growth is less than the cement production growth rate during the period. As expected, the activity effect (△P) is dominant factor for the increase of air pollutant emissions. The activity effect has a positive driving force to air pollutant emissions, contributing to the increase of PM emissions from 788 kt to 4022 kt, and the increase of SO_2 emissions from 152 kt to 904 kt, and the increase of NOx emissions from 203 kt to 1434 kt from 2006 to 2012, respectively. The energy intensity effects (△EI) offset the increase of air pollutant emissions, which accounted for 31.5% of SO_2, 24.7% of NOx, and 23.2% of PM emissions in 2012. Furthermore, the effects of emission factors (△EF) heavily depend on the implementation status of air pollution control technologies and thus widely vary among pollutants. For PM and SO_2 emissions, the first emission standard of air pollutants for cement manufacture was carried out in 1996, which brought

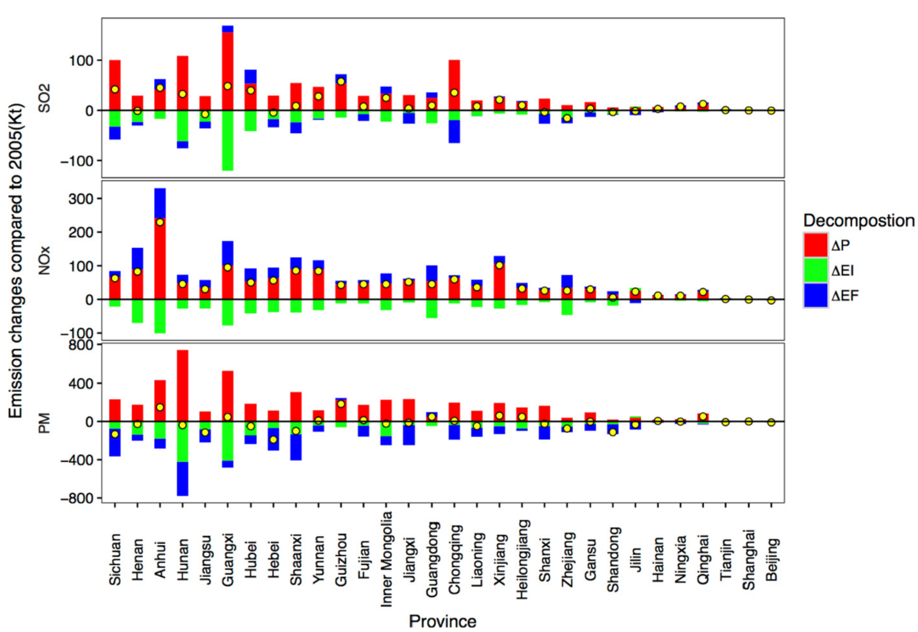

Fig. 8. LMDI decomposition analysis of SO_2, NOx and PM emissions from cement industry in 2012 in 30 provinces compared to 2005. Note: (1) Yellow dots denote the overall emission change from 2005 to 2012. (2) the provinces are ordered by the cement production increment from 2005 to 2012, from the largest increment in Sichuan and the lowest increment in Beijing. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

profound reductions in emissions afterwards. While the emission standard for NOx was issued in 2004, which resulted in a slower adoption of NOx control technologies compared to PM and SO_2 control technologies (Ministry of Environmental Protection of China (2004); Ministry of Environmental Protection of China (1996)). The adoption of air control technologies will further decrease air pollution, especially for the western region of China, with the implementation of the new standard of air pollutants for the cement industry in 2013 (Ministry of Environmental Protection of MEP of China, 2014).

To illustrate the features of the heterogeneity of emissions of air pollutants among provinces, Fig. 8 shows the proportion (including activity effect, energy intensity effect, and emission factor effect) of emissions of SO_2, NOx, and PM on the provincial level, based on LMDI decomposition analysis. As expected, production is driving emissions up, while improvements in energy efficiency tends to, ceteris paribus, reduce the emissions. The contributions of production and energy efficiency to air pollutant emissions differ widely among provinces. The production effect had the largest contribution to the changes of SO_2 emissions in Guangxi, followed by Sichuan, Hunan, and Guangdong, respectively. The activity effect ranked the largest contribution to the changes of NO_x emissions in Anhui, followed by Guangxi and Henan. The effects of energy intensity and emission factors play an important role in offsetting the increments of total air pollutant emissions in all provinces. Provinces located in the east and south of China contributed most to the offsetting of PM and SO_2 emissions. Guangxi and Anhui were the top provinces in mitigating emissions of SO_2, NOx, and PM, respectively. All driving factors had limited impact on the trends of air pollutant emissions in developed regions (e.g. Beijing, Shanghai, Tianjin) and developing regions with small contributions to total national cement production (e.g. Hainan and Ningxia).

5. Conclusion

China is the largest cement producer worldwide and is one of the largest energy consumers and emitters of CO_2 and air pollutants. The objective of this study is to provide insights for understanding the key driving factors that influence the changes of energy consumption and emissions of CO_2 and air pollutants in China's cement industry on national and provincial scales through employing spatial-temporal decomposition analysis. Previously, the provincial heterogeneity of the effects of different diving factors was scarcely assessed. China's CO_2 emissions from the cement industry has increased rapidly in the period of 2005–2012, driven by the growth in cement and clinker production. The emission increases were partly offset by decreases in energy intensity through energy efficiency improvements and fuel substitution in the Anhui, Jiangsu, and Sichuan, i.e. provinces with high cement production. At the provincial scale, the energy intensity and emission factor affect vary greatly in the provinces but much less in the future.

Before 2009, the emissions of SO_2, NOx, and PM in China's cement industry declined significantly, because more stringent air pollution standards were implemented. After 2009, air pollutant emissions started to rise with high growth rate in cement production, and were only slightly offset by reductions from energy intensity improvements. At the provincial scale, in the period 2005–2012, the developing regions, such as Anhui, Shandong and Sichuan had a great share of total air pollutant emissions, while the megacities of Beijing, Shanghai, and Tianjin contributed less to the total emissions. The trends reflect different trajectories between developed and developing regions. The activity effect (i.e., the production of clinker and cement) played an important role in air pollution abatement in the future but have large variations among provinces. The effects of energy intensity and emission factor had limited impacts on the emission trends in developed regions (e.g. Beijing, Shanghai, Tianjin, etc.). The findings in the study may be relevant to narrow down the projection of GHG mitigation and air pollution abatement on economic and technical perspectives.

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